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Between Dry Rock and a Salty Place: Freshwater Species in a Coastal Environment Facing Climate Change

Natasha Viadero
Florida International University, nviad001@fiu.edu

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FLORIDA INTERNATIONAL UNIVERSITY

Miami, Florida

BETWEEN DRY ROCK AND A SALTY PLACE:

FRESHWATER SPECIES IN A COASTAL ENVIRONMENT FACING CLIMATE CHANGE

A Thesis submitted in partial fulfillment of
the requirements for the degree of
MASTER OF SCIENCE
in
ENVIRONMENTAL STUDIES
by
Natasha Marie Viadero

2022

To: Dean Michael R. Heithaus
College of Arts, Sciences, and Education

This thesis, written by Natasha Marie Viadero, and entitled Between Dry Rock and A Salty Place: Freshwater Species in a Coastal Environment Facing Climate Change, having been approved in respect to style and intellectual content, is referred to you for judgment.

We have read this dissertation and recommend that it be approved.

Elizabeth Anderson

Micheal Allen

Rolando Santos, Co-Major Professor

Jennifer Rehage, Co-Major Professor

Date of Defense: November 1, 2022

The thesis of Natasha Marie Viadero is approved.

Dean Michael R. Heithaus
College of Arts, Science and Education

Andrés G. Gil
Vice President for Research and Economic Development
and Dean of the University Graduate School

Florida International University, 2022

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ABSTRACT OF THE THESIS

BETWEEN DRY ROCK AND A SALTY PLACE: FRESHWATER SPECIES IN A COASTAL ENVIRONMENT FACING CLIMATE CHANGE

by

Natasha Marie Viadero

Florida International University, 2022

Professor Jennifer Rehage, Co-Major Professor

Professor Rolando O. Santos, Co-Major Professor

Numerous species face redistribution and compression of habitat due to climate change. We coupled long-term movement and environmental data to assess how a freshwater species responds to changes in a coastal refuge habitat using Boosted Regression Trees (BRT) to predict distributional changes in the coming decades. Salinity, variation in salinity, and stage of the surrounding marsh habitat were the most important variables in BRT model, accounting for over half (56.6%) of the tree splits informing the final model. Interestingly, the habitat classified as conditional experienced the most variability (5.85 ± 6.2 km^2), while core habitat remained relatively consistent (1.29 ± 0.98 km^2) across years with varying hydrological conditions. These results suggest that varying environmental scenarios can drastically shift the amount of suitable habitat available for freshwater species using high salinity, conditional habitats at the coast. Climate change will likely result in large-scale reductions of critical dry season habitat for these species; while restoration efforts and adaptive management can bolster the resiliency of these habitats to ensure population persistence.

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ABBREVIATIONS AND ACRONYMS

AIC	Akaike Information Criterion
BRT	Boosted Regression Trees
ENP	Everglades National Park
FCE	Florida Coastal Everglades
LTER	Long Term Ecological Research
ML	Machine Learning
PPT	Parts Per Thousand
SR	Shark River
SLR	Sea Level Rise
TL	Total Length

CHAPTER I

GENERAL INTRODUCTION

Climate change has been at the forefront of scientific inquiry due to the uncertainty surrounding its severity as well as impacts on economies and ecosystems. Increased global temperature, reductions in glacial coverage, ocean acidification, sea level rise, increased frequency of extreme climate events, and altered precipitation regimes are just some of the consequences of climate change that have already been documented (Alexander et al. 2014; Malanson and Alftine 2016; IPBES 2019). However, these climate change impacts are not equally distributed leaving some areas more vulnerable than others. Research aimed at scaling down the General Circulation Models to be more informative for management planning in South Florida reveals that the projected median increase in temperature for the region is around 1 – 1.5 °C within the next 50 years (Obeysekera et al. 2014). There is more uncertainty regarding changes in precipitation (-20 to +10%), but suggest a reduction in rainfall for the region (Obeysekera et al. 2014). Projected increases in temperature and associated evapotranspiration rate coupled with decreased precipitation leave South Florida, the Everglades, and restoration plans highly vulnerable to climate change.

The lack of elevation, extensive coastline, and historic fragmentation of southward freshwater flow makes the Everglades ecosystem particularly susceptible to sea level rise (SLR) and saltwater encroachment. Fortunately, the ongoing Comprehensive Everglades Restoration Plan (CERP) is the largest ecosystem restoration project undertaken in the United States and has the potential to minimize the impacts of SLR to Everglades National Park (U.S. Army Corps of Engineers 2020). Of primary importance is CERP's focus on restoring historic southward freshwater flow into Everglades National Park (ENP) to replenish groundwater, mitigate hypersalinity events in Florida Bay, and enhance habitat for fish, wildlife, and recreation (National Academies of Sciences, Engineering, and Medicine 2021). Research investigating how ENP is effected by SLR and water management show a trend towards a more saline environment, specifically in years with lower upstream freshwater inflow, and a 13-cm increase in sea level between 2001 and 2016 (Dessu et al. 2018). Furthermore, Everglades simulation models for varying climate change scenarios show the saltwater front has the potential to move up to 15 km inland beyond its current position within 40 years altering the distribution of freshwater habitats (Flower et al. 2017). These projections raise the question as to whether restoration efforts will sufficiently counteract the impacts of SLR on ENP's native species.

Of particular interest is the ability of freshwater fish species to cope with a saltier and potentially drier Everglades. Salinity is considered to be a key environmental factor in the growth and survival of fish due to its influence on physiological processes (Bachman and Rand 2008; Castillo et al. 2018). Furthermore, even periodic pulses of high salinity are thought to act as strong selective forces on freshwater fish (Norris et al. 2010). To reduce their exposure to salinity, the distribution of freshwater species is likely to change in response to SLR with subsequent effects on ecological processes through altered food webs, reduction in species richness and abundance, and ultimately scale-up to influence ecosystem functions and services (Malanson and Alftine 2016; Pecl et al. 2017). A clear understanding of how salinity influences the distribution of freshwater fish species in ENP is essential for evaluating how they may respond to SLR and saltwater encroachment.

One of these key freshwater fish species is the Florida Largemouth Bass (*Micropterus salmoides floridanus*). Florida Largemouth Bass, hereafter referred to as Bass, are an ecologically and economically important species throughout their life histories (Taylor et al. 2019). They are highly prized among the fishing community due to their fighting ability and fast growth rate. Due to their popularity in the angling community, management agencies have introduced Bass to waters throughout the United States, Mexico, South America, Europe, and South Africa through stocking programs making them one of the most targeted species worldwide (Kim et al. 2022). In North America, Bass were estimated to be the target species of 90% of fishing tournaments between 2009 and 2011 in the Southeast region of the United States (Driscoll et al. 2012). Throughout the Greater Everglades, they are the most recreationally targeted species and contribute an estimated $141 million annually to Florida's economy accounting for 40% of the economic contribution of Everglades freshwater fishing (Fedler 2009). Being a freshwater species, Bass are susceptible to physiological stress when exposed to salinity. Although they are capable of withstanding relatively high salinities for a short time, prolonged exposure (120 days) to 12 parts per thousand (ppt) can be fatal and a marked decline in abundance is observed when salinities temporarily exceed 16 ppt (Meador and Kelso 1990; Glover et al. 2012).

The resilience of this recreational fishery relies on healthy fish populations and on maintaining favorable environmental conditions in the ecosystems that support them. Though there is mounting evidence of saltwater encroaching further into coastal systems as a result of SLR, our understanding of how SLR and saltwater encroachment may limit habitat availability for Bass remains limited. Additional information on what drives habitat use in these vulnerable areas will only strengthen our understanding of the system and deepen our appreciation for the Everglades. This work addresses how environmental conditions influence the habitat use of Bass in SR and to what degree their dry season refuge habitat fluctuates under varying environmental conditions. Ultimately, developing predictive frameworks for the behavioral responses of ecologically and economically valuable species under varying abiotic conditions is critical to the establishment of successful management and restoration practices in the face of climate change.

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CHAPTER II

BETWEEN DRY ROCK AND A SALTY PLACE: FRESHWATER SPECIES IN A COASTAL ENVIRONMENT FACING CLIMATE CHANGE

INTRODUCTION

Climate change is driving large-scale shifts and redistributions of species across the globe (Weiskopf et al. 2020). These shifts in distribution, abundance, behavior, and phenology can often lead to unanticipated negative effects on ecosystem structure, function, and services (Pecl et al. 2017; IPBES 2019). Although widespread, responses to climate change vary relative to variation among taxa in exposure, physiological tolerance, and coping ability (Staudinger et al. 2013; Beever et al. 2016). Populations that occur near or at their physical or physiological limits are particularly vulnerable to even the slightest changes in the ecosystem that supports them. Mountain top species, for example, have been found to shift 100 meters uphill for every one-degree Celsius increase in temperature to remain in their physiological range (Freeman et al. 2018). Physiologically restrained at their downhill boundary and physically limited by the mountain peak, these species are experiencing *compression* of suitable habitat. By compression, we refer to the reduction in the spatial extent of a given habitat (Brown et al. 2016). This leaves researchers, resource managers, and conservation practitioners with the pressing need to understand the degree of habitat compression experienced by taxa and how active management could mitigate those impacts (Henry and Sorte 2022).

Recent research shows considerable habitat compression for estuarine-dependent species under a variety of climate change scenarios. For the Delta Smelt in California, increasing temperatures are likely to cause thermal habitat compression within 50 years (Brown et al. 2016). Marinization of the Swan–Canning Estuary in Western Australia has been associated with the decline of flathead grey mullet (*Mugil cephalus)* whose juveniles prefer fresh conditions (Valesini et al. 2017). This highlights the pressing need to evaluate how species respond to variation in environmental conditions that drive habitat suitability to better predict effects of climate change. This is particularly essential for estuaries, where gradients in environmental conditions create diverse habitats which support a variety of species and provide numerus ecosystem services and functions (Brooker and Scharler 2020). Within estuaries, the spatial extent and distribution of various habitats are dependent upon interactions between freshwater inputs and tidal processes. Further, estuaries and tidal rivers are experiencing the most changes in tidal trends due to climate change-related alterations in timing and availability of freshwater and sea level rise (Talke and Jay 2020; Khojasteh et al. 2021), leaving them highly susceptible to habitat compression that threatens population persistence.

Assessing distributional shifts and habitat compression in aquatic species has long posed challenges for researchers and resource managers (Jeltsch et al. 2013). The rapid advancement of acoustic telemetry, however, provides a unique ability to passively monitor the movements of aquatic species to help inform habitat use and suitability (Nathan et al. 2022). Acoustic telemetry uses implanted acoustic transmitters and an array of acoustic receivers that 'listen' for the uniquely identified transmitters, or tags (Hussey et al. 2015). This capacity to obtain data on numerous species in a variety of aquatic settings has transformed movement ecology into a data-rich field, with high spatiotemporal resolution of an animal's distribution and habitat use (Nathan et al. 2022). As such, acoustic telemetry has provided insight on migration patterns, predator-prey interactions, and has restructured our understanding of aquatic environments (Daniels et al. 2018; Boucek et al. 2019). In particular, the high spatiotemporal resolution of acoustic telemetry creates opportunities to assess responses to both acute disturbances and/or gradual environmental changes (Massie et al. 2020; Matley et al. 2022). For example, Crear et al. (2020) showed that the timing of arrival and departure of Cobia in Chesapeake Bay is temperature-dependent, and climate change is likely altering this migration phenology (Crear et al. 2020). Coupling long-term acoustic telemetry and environmental variables allows researchers to address questions about distributional shifts expected with climate change, particularly for areas with high inter- and intra-annual variations in environmental conditions.

Everglades National Park (ENP), in South Florida, experiences high levels of interannual variability in environmental conditions, stemming from fluctuations in precipitation, water management, climatic oscillations (El Niño, La Niña, and Atlantic Multi-decadal Oscillation), and extreme weather events such as drought and hurricanes (Davis et al. 2018; Abiy et al. 2019). The availability of freshwater habitat fluctuates with seasonal rainfall and drives the distribution and abundance of freshwater fish species (Ruetz et al. 2005; Chick et al. 2006; Goss et al. 2014). As the dry season progresses, water levels recede resulting in a complete dry-down of marsh habitat for most years. This higher dry down frequency is the result of drainage, impoundment and wetland loss across the ecosystem (Marshall et al. 2020). These seasonal dry-downs force freshwater species to seek dry season refuge in coastal habitats where they are vulnerable to physiological stress related to salinity exposure. The lack of elevation, extensive coastline, and historic fragmentation of coastal freshwater flow makes the Everglades ecosystem particularly susceptible to sea level rise (SLR) and saltwater encroachment (Dessu et al. 2021). This raises the question of how freshwater fish species may respond to potentially increased seasonal or year-around compression of freshwater habitat expected with climate change. Everglades simulation models for varying climate change scenarios show the saltwater front has the potential to move up to 15 km inland beyond its current position within 40 years altering the distribution of freshwater habitats (Flower et al. 2017).

In this study, we coupled long-term movement and environmental data to assess how a freshwater fish species (Florida Largemouth Bass, *Micropterus salmoides floridanus*) responds to variation in environmental conditions in an Everglades estuary, in order to better elucidate expected future coastal habitat compression with SLR. In particular, we asked the following questions: (1) what environmental variable(s) determine dry season habitat availability in the estuary? and (2) how does the spatial extent of available habitat change under varying environmental conditions? These questions allow us to determine the driving variables and extent of present habitat compression over the dry season, in order to elucidate future habitat compression expected with SLR, which may be counteracted by restoration efforts. We hypothesized that species spatial distribution is limited by physiological constraints and habitat availability, and consequently, habitat compression fluctuates yearly with severity of marsh drying. We used Boosted Regression Tree (BRT) analysis to identify the most important environmental variables influencing habitat suitability following Bangley et al. (2018). We then mapped and classified model outputs into four habitat types and assessed fluctuations in the spatial extent of habitat types among years of varying environmental conditions.

METHODOLOGY

Focal Species

Florida Largemouth Bass (*Micropterus salmoides floridanus*), hereafter referred to as Bass, were first described as a subspecies of black bass by Bailey and Hubbs (1949). Although somewhat contested, they have been genetically differentiated from their sister species, the Northern Largemouth Bass (Barthel et al. 2011; Taylor et al. 2019; Kim et al. 2022). Ecologically and economically important throughout their life histories, juveniles are an important prey source for various wading birds and other predatory fish species while adults feed relatively high in the food web and provide top-down control (Boucek & Rehage, 2013; Taylor et al. 2019). Bass are highly-prized among the fishing community due to their fighting ability and fast growth rate. After overharvesting led to extinction concerns, voluntary catch-and-release practices were widely adopted by anglers (Taylor et al. 2019). In North America, Bass are the most targeted freshwater species by anglers and as such, contribute significantly to local economies through gear purchases, tournament fees, guide fees, and restaurant meals (Siepker et al. 2007; Fedler 2009; Kim et al. 2022). Being a freshwater species, Bass are susceptible to physiological stress when exposed to salinity. Although they are capable of withstanding relatively high salinities for a brief time, prolonged exposure (120 days) to 12 parts per thousand (PPT) can be fatal, and a marked decline in abundance is observed when salinities temporarily exceed 16 PPT (Meador and Kelso 1989; Glover et al. 2012). As generalists, Bass are considered proxies for ecosystem health metrics, and for other freshwater species with similar physiological limitations (Kemp et al. 2019). Given their ecological and socioeconomic importance and physiological limitations, Bass are an ideal candidate for evaluating the response to environmental conditions in an area that is highly susceptible to the effects of climate change.

In the Everglades, Bass are forced to move into deep-water refuge as the dry season progresses and marsh habitats become unavailable (December – May; Boucek and Rehage 2013). Coastal habitats provide much of this critical dry-season refuge (Pierce et al. 2020). However, the amount of freshwater habitat in these coastal environments is driven by interactions between freshwater inflow and tidal processes. This dynamic salinity regime presents Bass with the challenge of finding suitable habitat when marshes are not available (i.e., dry-down), particularly in low freshwater inflow years where freshwater habitat in the estuary is also limited by the increasing salinity. As the dry season progresses and salinity pushes further upstream into the coastal riverine system, suitable habitat for freshwater species is reduced, and Bass are confined between dry marsh habitat and rising salinities until marshes are reflooded with the onset of the wet season, typically in June (Pierce et al. 2020) The compression in available habitat forces Bass to cope with stressful conditions and, during years of extreme drought, can cause mortality (Pierce et al. 2020). Considering altered precipitation patterns and SLR projections for the area, these events are likely to increase in frequency (Hall et al. 2019).

Bass in the Shark River

The Shark River (SR) in western ENP connects graminoid freshwater marshes upstream to extensive mangrove forest downstream and acts as ENP’s main drainage (Dessu et al. 2018, Figure 1a). Permanently inundated, the river provides the largest deep-water refuge for species driven out of the surrounding drying marsh and supports a substantial recreational Bass fishery (Boucek & Rehage 2013; Pierce et al. 2020). Along its 32 km length, SR transitions from ecotonal mangrove creeks to a brackish estuary which ultimately connects to the marine waters of the Gulf of Mexico. The upper portion of SR, or Upper SR, generally maintains salinities below 3 PPT. Tarpon Bay, a shallow, relatively open water bay at the midpoint of Shark River (located at river km 15 - 23), has an annual mean salinity of 5 PPT with seasonal salinity fluctuations between 1 PPT in the wet season and 20 PPT in the dry season (Gaiser & Childers 2020), and is the lower most part of the river used by Bass. During years of extreme drought (i.e. low freshwater inflows), salinities can rise above 10 ppt in the headwaters and approach 30 ppt in Tarpon Bay.

Acoustic telemetry in Shark River

The Shark River Acoustic Array has been tracking the movement of several consumer species since 2009 (Boucek et al. 2017; Massie et al. 2020; Strickland et al. 2020). Bass tagging efforts began in 2013 on a pilot basis and expanded in subsequent years. Bass greater than 40 centimeter (cm) in total length (TL) were captured, measured, weighed, and set in a cooler filled with ambient river water in preparation for tag implantation (Table A.1, Appendix). Internal implantation of an acoustic tag is the most common method for fish tagging since the high retention rate often outweighs the potential for tagging mortality from the procedure (Wagner et al. 2011). A ventral incision roughly 30-mm in length is made to allow implantation of an acoustic transmitter (69 kHz V13, Innovasea, Halifax, NS, Canada) into the abdominal cavity (Massie et al. 2020). The incision is sealed with one suture at the mid-point of the opening and Bass are allowed to fully recover before being released at the capture location (IACUC-20-057-AM02). Acoustic receivers, or stations, remain submerged at fixed locations throughout the river and as a tagged individual swims within range of an acoustic receiver the tag code, time, and date are recorded. The full array consists of 41 acoustic receivers; however, this work focuses on the upstream portion of the array where all bass detections are contained (River Km 15 to 32, Figure 1a). Salinity and temperature HOBO data loggers (U24-002-C) were deployed at a subset of receiver locations to supplement 5 permanent hydrostations (Figure 1a) from which environmental data could be obtained. Over a total of 8 years, bass were detected in the upper 21 receivers of the array, with a higher frequency of detections at the upstream most receivers (Figure 1b).

Environmental Data

Focal environmental data used in analysis included: salinity, temperature, and river depth with the first two variables obtained from seven HOBO loggers and five hydrostations (Figure 1a). Marsh stage was obtained from the nearest hydrostation upstream of the river, MO215 (Figure 1a). Depth was measured at receiver locations. HOBO loggers were deployed to increase the spatial resolution of salinity and temperature data; however, the deployment did not cover the full study period (2014-2021). HOBO loggers were deployed at seven receiver locations in March of 2021 and recovered in February of 2022. To rectify the disparity in temporal resolution, HOBO logger data was related to the long-term hydrostations through linear regressions in R. Fitted equations were developed for each HOBO logger through an iterative process, checked for collinearity, and selected based on the Akaike information criterion (AIC). If there was no difference in AIC, the most parsimonious model was selected. Fitted equations from the final models were then used to extrapolate salinity and temperature at HOBO logger locations for the full study period (Table A2 & A3 in Appendix).

Boosted Regression Trees

BRT is a powerful Machine Learning (ML) statistical technique that provides advantages over more traditional methods for describing complex ecological processes (Dedman et al. 2015). BRT combines the power of regression and boosting to form a single final model. Regression trees identify breaks in the probability of presence or absence of a target species along the range of values of each variable included in the analyses (Bangley et al. 2020). Boosting is a stepwise process where each subsequent tree is informed by the previous tree until a minimum deviance between models is reached. Boosting is based on the concept that averaging many rules is more accurate than a single prediction rule (Elith et al. 2008). BRT are robust against multicollinearity, outliers, and zero-inflated data, and are capable of fitting nonlinear relationships common in ecological data (Elith et al. 2008).

Sample data

All data preparations were completed with RStudio 2021.9.0.351 (R Core Team, 2018). Acoustic telemetry data was pulled for all receivers that met one of the following criteria: the receiver had a HOBO logger attached (n = 7), or the receiver was located within 500 meters (n= 3), the mean detection range of receivers in SR, of a permanent hydrostation. This resulted in a total of 10 receivers used in the analysis (Figure 1). Following Bangley et al. (2020), we extracted presence to then relate to environmental variables. Presence was defined as the detection of at least one tagged Bass at a given receiver on a given day (i.e., daily detected/not detected is the response variable in models). This data transformation corrects for spatial and temporal autocorrelation (Bangley et al. 2020). A data matrix of daily observations for all eight years was then created combining the detection data (0 and 1 for not detected or detected, respectively) and the environmental data for the 10 receivers included in the analysis. The variables included in the model (all at a daily scale) were: mean salinity (PPT) and temperature (°C), daily standard deviation of salinity and temperature, depth (m), Julian day, and upstream marsh stage (cm). Variables were selected based on the following hypothesized relationships: low levels of marsh stage force Bass into dry season refugia (marsh stage), salinity limits the spatial extent of estuarine use (mean salinity), higher temperatures are unfavorable for a temperate species (mean temperature), Bass prefer more stable environments (standard deviations of salinity and temperature), Bass are associated with shallow environments (depth), and Bass immigration and presence in the estuary may be responding to a fixed variable and not environmental conditions (Julian day). All variables were specific to one of the 10 receivers used in the analysis, apart from Julian day and marsh stage which were held constant across all receivers. This resulted in a matrix 25,550 of daily rows used for analysis across the eight years of movement data.

BRT modeling for Bass presence in the Shark River

BRT were constructed using the gbm.auto package in R created by Simon Dedman, which automates the boosted regression process and generates model outputs (Dedman et al. 2017). In our models, we use BRT to classify presence/absence across the

10 receivers as a function of the seven predictor variables described above. Model outputs included relative importance and marginal effect plots for each variable included in the model, cross-validation results, area-under-the-curve scores, common Machine Learning (ML) evaluation metrics, and probability of occurrence (see additional details in Elith et al. 2008 and Dedman et al. 2017). ML evaluation metrics, which include true positive, true negative, false positive, and false negative classification rates, provide information on the overall accuracy of the model. The main function, gbm.auto, allows the user to include different combinations of learning rates, tree complexities, and bag fractions, all of which are tested iteratively. Learning rate determines the contribution of each tree to the model, tree complexity allows for interaction among predictor variables, and bag fraction adjusts the proportion of data used to train and test the model. Training and testing data were used for cross-validation and to produce ML evaluation metrics. Learning rates of 0.005 and 0.01, tree complexities of 2, 4, and 6, and bag fractions of 0.5 and 0.75 were tested. Lower learning rates increase the number of trees required to formulate the final model and are dependent on the number of observations included (Elith et al. 2008). Tree complexities and bag fractions were selected following Bangley et al. (2018) and Dedman et al. (2017). Bag fractions of 0.5 split the data evenly into training and testing datasets, while a bag fractions of 0.75 use 75% of the data to train the model and 25% to test it.

Predicting occurrence in the Shark River

Relationships between Bass presence and environmental conditions were then used to predict and map the probability of occurrence of Bass in the study domain in a spatially-explicit manner (River km 15 to 32, Figure 1) as a function of environmental conditions across the eight years of the study. We restricted our prediction grids to dry season months (January – May) since Bass primarily use SR as dry season refuge (Boucek and Rehage 2013b; Pierce et al. 2020) which was confirmed by our detection data. Across the eight years of detection data, 71% was collected between January and May. Following Bangley et al. (2018), Bayesian Empirical Kriging in ArcGIS Pro was used to interpolate all location-specific environmental variables, rasterized at 500-meter area grids to match the mean detection range of SR receivers (unpublished data), and extracted for each month analyzed. This process was repeated for all eight years of the study period.

The model predicts the probability of occurrence (0 –1) at each 500m grid cell for each set of environmental conditions. In other words, the likelihood of detecting a Bass under specific conditions at that cell. The probability of occurrence for each month and year combination were classified into four groups, unsuitable (< 0.01), marginal (0.01 – 0.1), conditional (0.1 – 0.5), and core (> 0.5). The total area for each classification was then calculated to identify what conditions produce the Maximum, Minimum, and Mean spatial extent of predicted occurrence and thus of available habitat across the eight years. The yearly sum of core and conditional habitat was used to select the three years used to evaluate how the spatial distribution of habitat changes under varying environmental conditions. The Maximum, Minimum, and Mean years were then mapped to visualize the predicted distribution of bass under varying conditions.

RESULTS

Environmental data regressions

Given that temperature is more uniform throughout SR, high levels of collinearity were evident among hydrostations; thus, fewer hydrostations were needed to inform the final temperature regression model for each HOBO logger (Table A.2 and A.3 in Appendix). Salinity at the two most downstream hydrostations also had high levels of collinearity, which were addressed in the same way as temperature. All final regression models of environmental data performed well with R^2 above 0.8. Across all seven HOBO logger stations, variance in both salinity and temperature data increased along a gradient from upstream to downstream, as expected, given greater marine influence and tidal amplitude downstream.

Factors driving bass presence in the Upper Shark River

The final BRT model parameters were as follows; a learning rate of 0.01, a tree complexity of 6 (number of tree nodes or length of each tree), and a bag fraction of 0.5. This 0.5 bag fraction, or 50/50 split of training and testing data corresponds to the recommended bag fraction by Elith et al. (2008). Overall, the model performed well with an overall classification rate of 0.94. More specifically, the model had a positive predictive power rate of 0.68 and a negative predictive power rate of 0.99 (Table A.4, Appendix). In other words, the model was able to classify roughly 70% of all positive observations (presence) and 99% of all negative observations (absence) correctly. The lower rate of positive predictive power suggests that the probability of occurrence produced by the model are likely more conservative than true values of occurrence.

Daily mean salinity was the most important predictor variable accounting for approximately 21.9% of the tree splits (Figure 2). The remaining variables in order of importance were the standard deviation of salinity (18.6%), marsh stage (16.1%), Julian day (13.8%), daily mean temperature (12.6%), depth (12.5%), and the standard deviation of temperature (4.5%). Marginal effect plots show an elevated probability of detecting the presence of bass at salinities below 10 PPT, at low standard deviation of salinity, and low marsh depths below < 30 cm (Figure 2). Marginal plots for the variables of lower importance (below 15%) are included as supplemental material (Figure A.1 in Appendix). The two largest interactions among predictor variables were between Julian day and temperature, and between Julian day and marsh stage, which was to be expected given the high seasonality of the system

Fluctuations in habitat occupied across years

As expected, the predicted total area of the estuary occupied by Bass varied across years and months of the study, and as a function of habitat classification (Figure 3 & Figure A.2 in Appendix). Area classified as core habitat where the probability of occurrence was greater than 0.5 (1.29 ± 0.98 km^2) showed less variance than the conditional habitat classification (5.85 ± 6.2 km^2), where the probability of occurrence was 0.1 - 0.5. Environmental conditions in 2014 characterized the mean total habitat area occupied by Bass across the eight years (Figure 4, second row). This mean year consisted of 1.13 km^2 of core habitat primarily detected in February and March, and a decreasing area of conditional habitat between January and May (from 1.23 to 0.16 km^2). Under these average conditions, at the onset of the dry season, much of the habitat in the Upper

SR is classified as conditional (since Bass are not consistently detected there) with portions transitioning to core habitat in February (Figure 4, second row and column). This pattern is largely driven by marsh conditions and the timing of marsh stage dropping below 30 cm (Figure 5c). The spatial coverage of core habitat remains relatively consistent between February and March, as salinity remains stable and low in the Upper SR (Figure 5a & Figure 5b).

Environmental conditions in 2018 provided the maximum total area occupied habitat with a nearly 4-fold expansion in areas classified as conditional relative to the mean across year. In 2018, in February and March, the total conditional habitat expanded to include the majority of Tarpon Bay (Figure 4, first row). This resulted in a total of 12.97 km^2 of conditional habitat, 4-times greater than the average of the time series. This expansion of habitat was created by a steadily decreasing march stage combined with persistent low salinities (30 cm of marsh stage and salinity maintained under 5 PPT, Figure 5). In contrast, 2015 produced the minimum total area occupied by Bass in the SR with core and conditional habitats contracting to less than half the spatial distribution of the mean year. This compression of habitat was due to severe and prolonged drying of marsh habitat (marsh stage below 30 cm for all five months) and subsequent increases in salinity throughout SR (Figure 5c and Figure A3).

DISCUSSION

As climate change intensifies, many species are experiencing a reduction in suitable habitat (i.e., habitat compression; Brown et al. 2016; Pecl et al. 2017). Here, we coupled acoustic telemetry and abiotic variables to better understand what environmental conditions drive the occurrence of a recreationally important freshwater species in a variable estuary. The environmental dependencies of Bass can provide resource managers with critical information in order to develop management targets and restoration goals that safeguard important dry season refuge for this and other freshwater species. Salinity, daily variation in salinity, and conditions in the surrounding marsh habitat were consistently the most important variables driving the occurrence of Bass in the Upper SR. Salinity below 10 PPT had the largest positive effect on Bass occurrence, relatively low levels of daily variation in salinity (<0.5 standard deviation) and marsh stage < 30 cm resulted in an increased probability of occurrence. Interestingly, the habitat that experienced the most variability was the conditional classification (5.85 ± 6.2 km^2), while the core habitat remained relatively consistent (1.29 ± 0.98 km^2) across varying conditions in the 8 years of the study. In other words, good conditions can generate 15.3 km^2 of core and conditional habitat for Bass, average conditions generate 4.4 km^2 of core and conditional habitat, and dry conditions compress Bass habitat down to 1.7 km^2 of SR. These results emphasize how varying hydrological conditions drastically shift the amount of habitat available to Bass. A deeper understanding how of Bass respond to environmental variables is critical for effective conservation and management of freshwater habitat in the face of climate change.

Daily salinity and daily variation in salinity were consistently the two most important variables in the BRT model. Generally, Bass are much less likely to use areas throughout SR with salinities above 10 PPT that experience oscillations in salinity associated with tidal processes. Our results suggest potential avoidance of salinity above 10 PPT prior to reaching the fatal level of salinity (12 PPT) found by Meador and Kelso (1990). Given that all Bass in this study were greater than 40 cm TL, it is important to note that our results may fail to capture size-specific tolerances recorded in other studies (Glover et al. 2013). Bass are most vulnerable to salinity during their egg and fry phase of life with 24% survival when exposed to salinity greater than 3.6 PPT (Tebo and Mccoy 1964). Although smaller individuals are capable of tolerating salinities above 3.6 PPT, the energetic cost of coping with elevated salinities increases with size (Glover et al. 2012; Kemp et al. 2019).

Salinity is known to be one of the most important driving factors of species distribution in estuaries (Glover et al. 2012; Castillo et al. 2018). Moreover, it provides an avenue for quickly evaluating the suitability of a system in near real-time and assessing a system’s ability to maintain critical habitat under a variety of climate change and SLR scenarios. For example, Everglades simulation models for varying climate change scenarios show the saltwater front has the potential to move up to 15 km inland beyond its current position within 40 years, altering the distribution of freshwater habitats (Flower et al. 2017). A 13cm increase in sea level between 2006 and 2016 has already been recorded for SR (Dessu et al. 2018). Using these projections and the results from this study, we can effectively calculate the habitat loss likely to be experienced by Bass and other freshwater species. At the same time, current restoration efforts in the region could mitigate this habitat loss by establishing salinity targets below 10 PPT for upstream portions on SR. Elevated levels of salinity in SR coincide with years of low freshwater inflow (Dessu et al. 2018).

Marsh stage is a major driver of ecological processes in the Everglades including faunal abundances and distributions (Goss et al. 2014). For instance, Parkos et al. (2011) showed that increased severity of drying was related to decreased catch per unit effort and shifts in community structure (Parkos et al. 2011). In our study, surrounding marsh conditions was the third most important variable explaining bass presence in the upper estuary. The steep increase in the probability of occurrence of Bass occurred when the marsh dropped below 30 cm in depth, which highlights the role of the SR as a dry season refuge. Bass are present in SR at the lower stages of marsh habitat during the dry season. Stages below 30 cm are typically detected in February - May. Thus, the timing of marsh drying and reflooding influences the timing of ingress and egress of Bass into SR. Our results suggest that resource managers may be able to manipulate marsh stage to control whether Bass remain in the surrounding marsh habitats. Additionally, our results indicate that a complete drying of the marsh is not necessary to trigger the movement of Bass into SR. Support for using Bass as an indicator species suggests that other freshwater species may follow the same pattern (Kemp et al. 2019) and that complete dry-down is unnecessary to maintain marsh-mangrove connectivity, prey-pulse dynamics, and a freshwater fishery in SR recorded in earlier studies (Boucek and Rehage 2013a; Pierce et al. 2020).

Further, previous work in the region indicates the survival probability of Bass is directly associated with the severity of marsh drying, with as low as 21% survival during years of drought (Pierce et al. 2020). Their results also showed that moderate levels of marsh drying (15 cm marsh stage) can increase survivability 3-fold when compared to severe levels of drying. Our results in combination with this previous work, then show Bass use SR when marsh habitats are unavailable. Ongoing tracking efforts shows a decline in the detection of Bass following the reflooding of marsh habitat suggesting a return to the surrounding marsh.

The SR is most susceptible to elevated salinities when freshwater inflows are low (Dessu et al. 2018). This effectively compresses suitable habitat for freshwater species, particularly toward the end of the season and results in greater compression in drought years, which can lead to population sink dynamics. Our results showed a 60% reduction of habitat during the Minimum coverage year (2015) when compared to the Mean coverage year (2014), resulting in a compression from 4.4 to 1.7 km^2 of habitat. Although SR Bass populations are capable of rebounding, successive years of low freshwater inflow and associated elevated salinities can be detrimental to the long-term persistence of Bass in the region. Our results support the need for increased water storage capacity in the region to help prevent severe, prolonged, and consecutive drying of the system. Uncertainty regarding regional precipitation patterns further emphasizes the need for increased water storage capacity (Obeysekera et al. 2014).

We saw marked differences in the spatiotemporal variation in Bass occurrence between core and conditional habitat (Figure 4). For core habitat, the main variance was temporal. This is most apparent in the minimum spatial coverage year (2014), where core habitat is present in January, minimal in February, and completely absent March – May. In contrast, in the average scenario, core habitat peaked in February and March. In the maximum scenario, core habitat is present in all months included in the analyses. Interestingly, conditional habitat experiences the most expansion and compression. This suggests that some portion of the population will expand their habitat usage to Tarpon Bay when conditions allow. Most concerning is the complete absence of core and conditional habitat in April and May of the minimum coverage scenario. During years of drought, Bass experience habitat compression and are forced to cope with unfavorable conditions until the surrounding marsh habitat refloods. This habitat compression gives rise to concerns regarding the long-term survival of Bass in SR. Past research suggests that to cope with elevated salinities, Bass shift energy allocation patterns from reproduction and growth towards increasing energy reserves (Norris et al. 2010). Moreover, additional population-level concerns arise from low survival rates associated with drought conditions and ensuing elevated salinities (Pierce et al. 2020).

The hydrology of the Everglades is driven by the seasonal precipitation patterns of the region (Price et al. 2008). Thus, changes in precipitation patterns as well as the frequency and intensity of extreme climate events related to climate change are likely to influence the overall health of the system (Alexander et al. 2014; Abiy et al. 2019). Variations in the amount and timing of freshwater poses serious challenges for resource managers as they attempt to balance increases in freshwater demand and climatic uncertainty. One way to develop resilience to these changes is by building water storage capacity. By storing excess water during wet periods, we can ensure the availability of freshwater during periods of severe drying (National Academies of Sciences, Engineering, and Medicine 2021). Moreover, our results provide actionable targets for resource managers regarding Bass in SR. Seasonal drying of marsh habitat is critical component of Everglades ecology by creating foraging opportunities for nesting wading birds and estuarine and marine predators (Beerens et al. 2011; Boucek and Rehage 2013a). However, a full drying of marsh habitat can be detrimental to freshwater species and is unnecessary to maintain foraging opportunities. Using 30 cm of marsh stage as a dry-down target can preserve foraging opportunities for wading birds while maintaining suitable habitat for Bass in SR (Bancroft et al. 2002).

With rising seas and reduced freshwater inflow, freshwater habitats in estuaries worldwide are experiencing habitat redistribution and compression that threatens the persistence of salinity intolerant species (Ficke et al. 2007; Arthington et al. 2010; Hallett et al. 2018). In SR, evidence shows a 13 cm increase in sea level rise between 2006 – 2016 (Dessu et al. 2018). Furthermore, climate change projections for the area show 15 km inland migration of the saltwater front (Flower et al. 2017). Without adequate management, critical dry season refuge habitat may be lost. Fortunately, restoration focused on increased freshwater flow is ongoing and has the potential to minimize the impacts of SLR on the region (U.S. Army Corps of Engineers 2020). Increasing freshwater storage capacity would ensure managers have the ability to prevent excessive drying of marsh habitat and mitigate salinity encroachment into freshwater habitats.

Adaptive management approaches are becoming increasingly important as we attempt to build resilience in the face of climate change and more traditional methods focused on maintaining historical conditions become less effective (Weiskopf et al. 2020). Our study shows that Bass expand and contract their use of SR according to environmental conditions, particularly salinity, daily variation in salinity, and surrounding marsh stage. SR may not be optimal habitat for Bass, but it provides critical dry season refuge. Availability of habitat with stable salinity below 10 PPT is critical when marsh stage falls below 30 cm and Bass are most likely to be using SR as dry season refuge. This work used Bass as an indicator species to provide water management targets for a broad range of freshwater species with similar physiological tolerances. A greater understanding of what drives the movement of freshwater species in an estuarine environment can provide insight on how they may respond to redistributions of habitat related to climate change. Clear and efficient metrics are vital for effective resource management (Weiskopf et al. 2020; Henry and Sorte 2022). Inadequate management of dry season habitat has direct and indirect influences on long-term survival, either through acute mortality events, or chronic reduction of fitness and fecundity. As such, the availability of dry season refuge habitat should be evaluated as it acts as a bottleneck for many species regardless of the duration of use. Climate change and sea level rise threatens the persistence of these vital freshwater dry season habitats and highlights the importance of adaptive management practices in future restoration efforts.

FIGURES

Figure 1. Map of full Shark River Acoustic Array in Shark River, Everglades National Park in Florida in black, and locations of environmental data loggers in blue and yellow (a.), focal study area with proportion of use by acoustic receiver in gray, environmental data loggers in black, and surrounding marsh habitat in light brown (b).

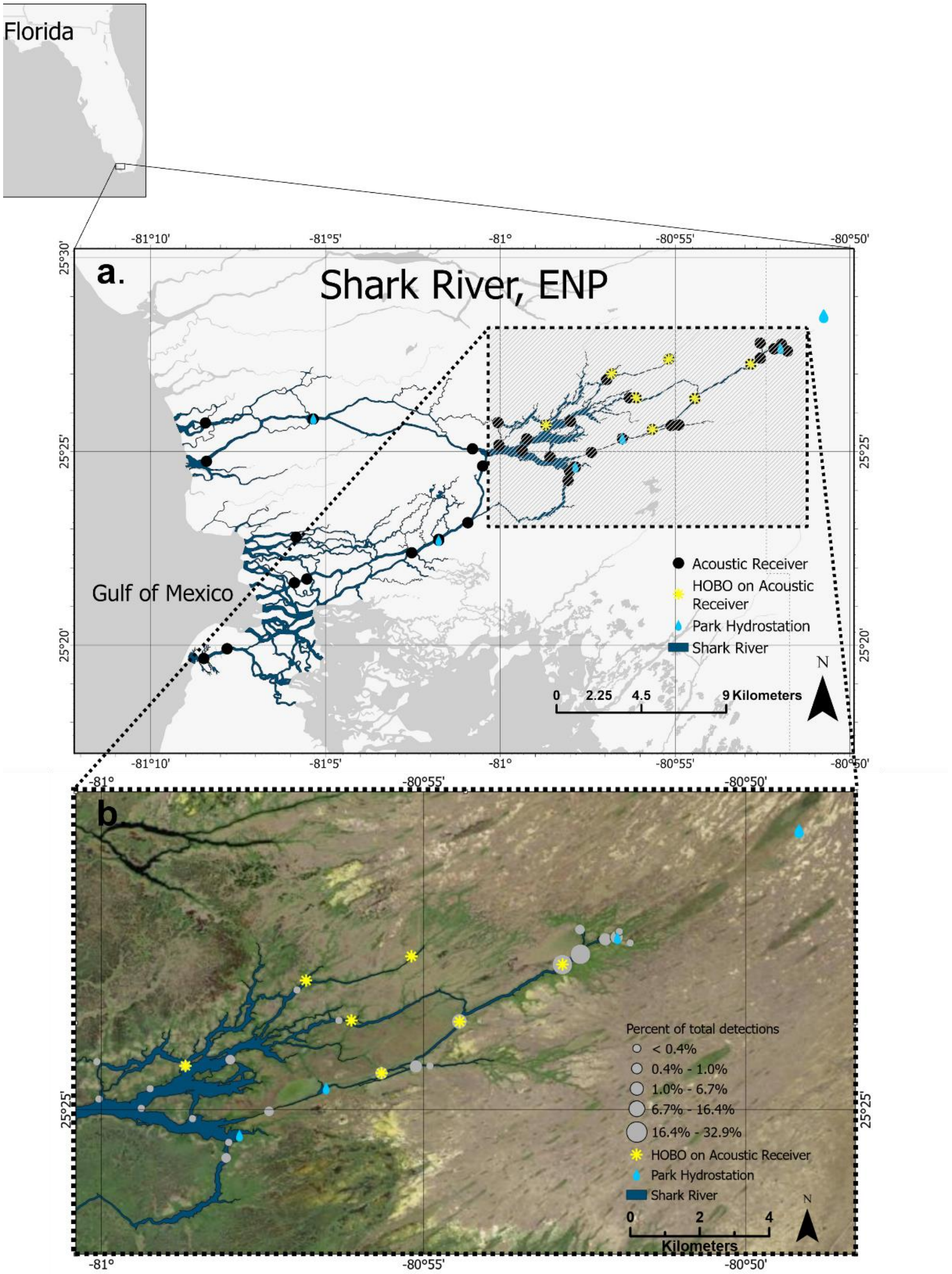


Figure 2. Marginal effect plots for the top three most important variables in BRT and their relative importance; salinity 21.8% (a), variation in daily salinity 18.9% (b), and marsh stage 16.1% (c).

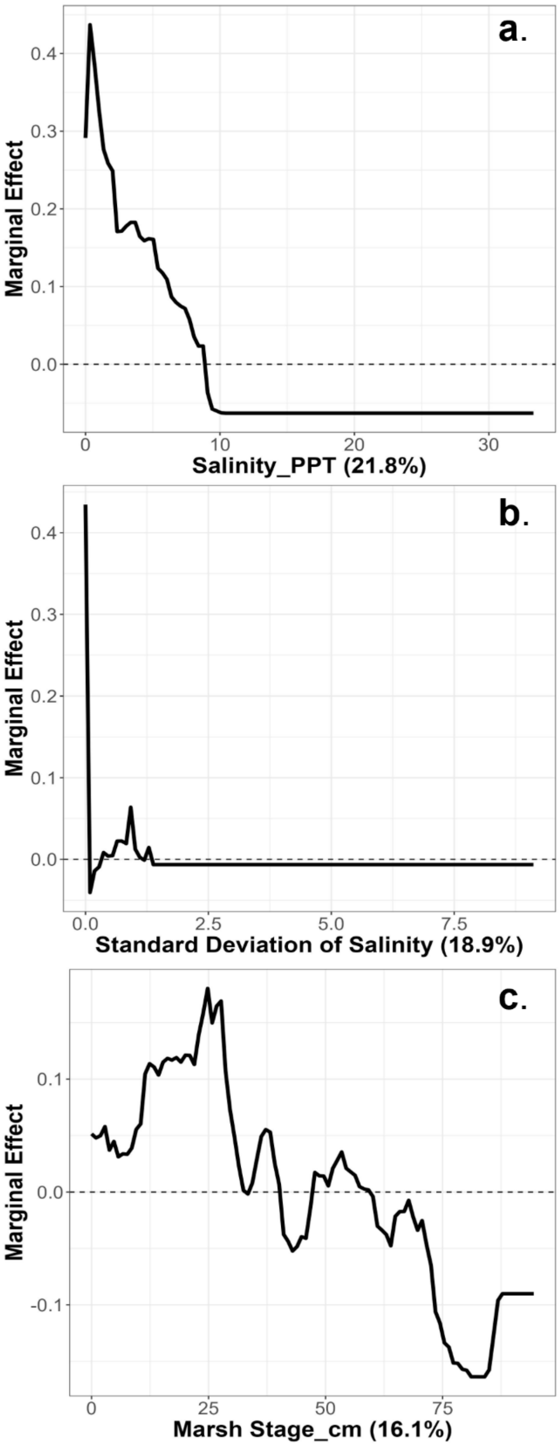


Figure 3. Total area classified as conditional (10-50%) and core (> 50%) by month and year. Years selected as Maximum, Minimum, and Mean years are highlighted by colored boxes.

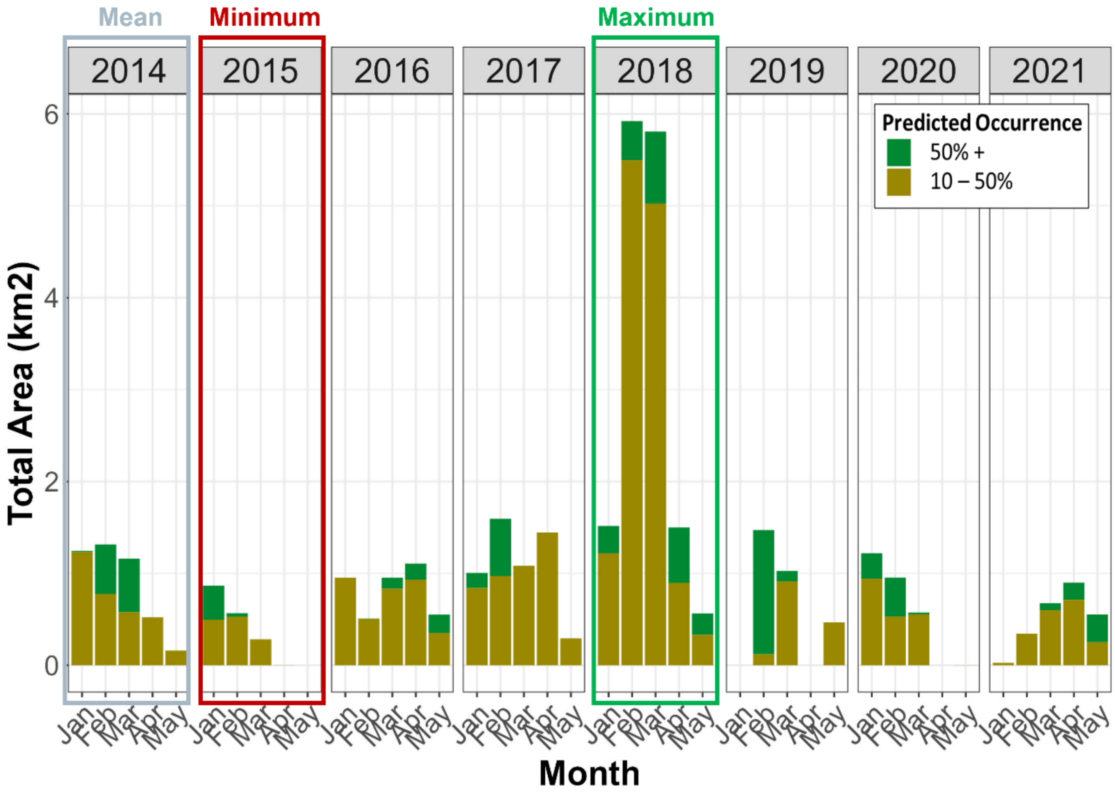


Figure 4. Florida Largemouth Bass habitat classifications based on BRT results by month to visualize the spatiotemporal variation under Maximum, Minimum, and Mean years.

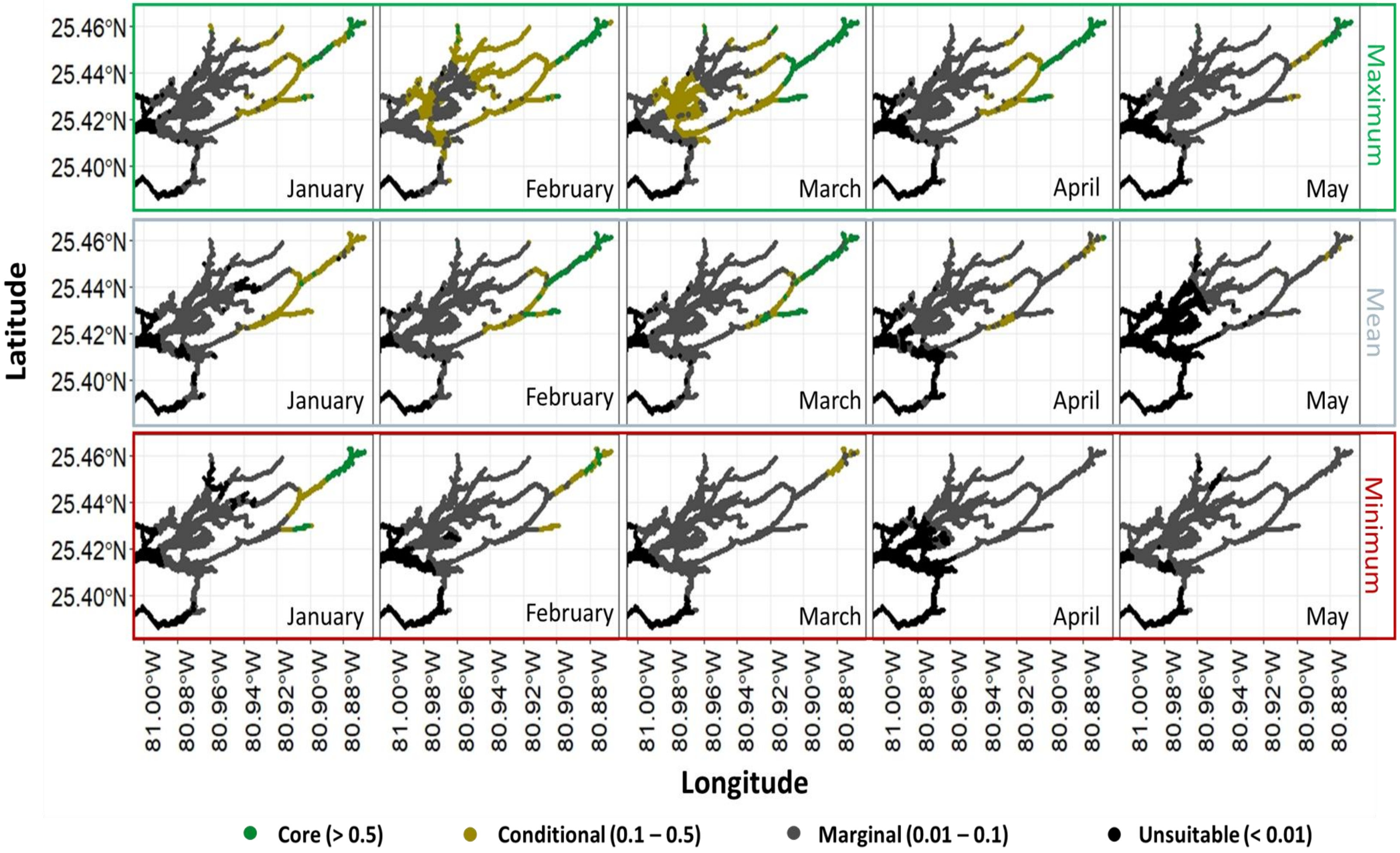
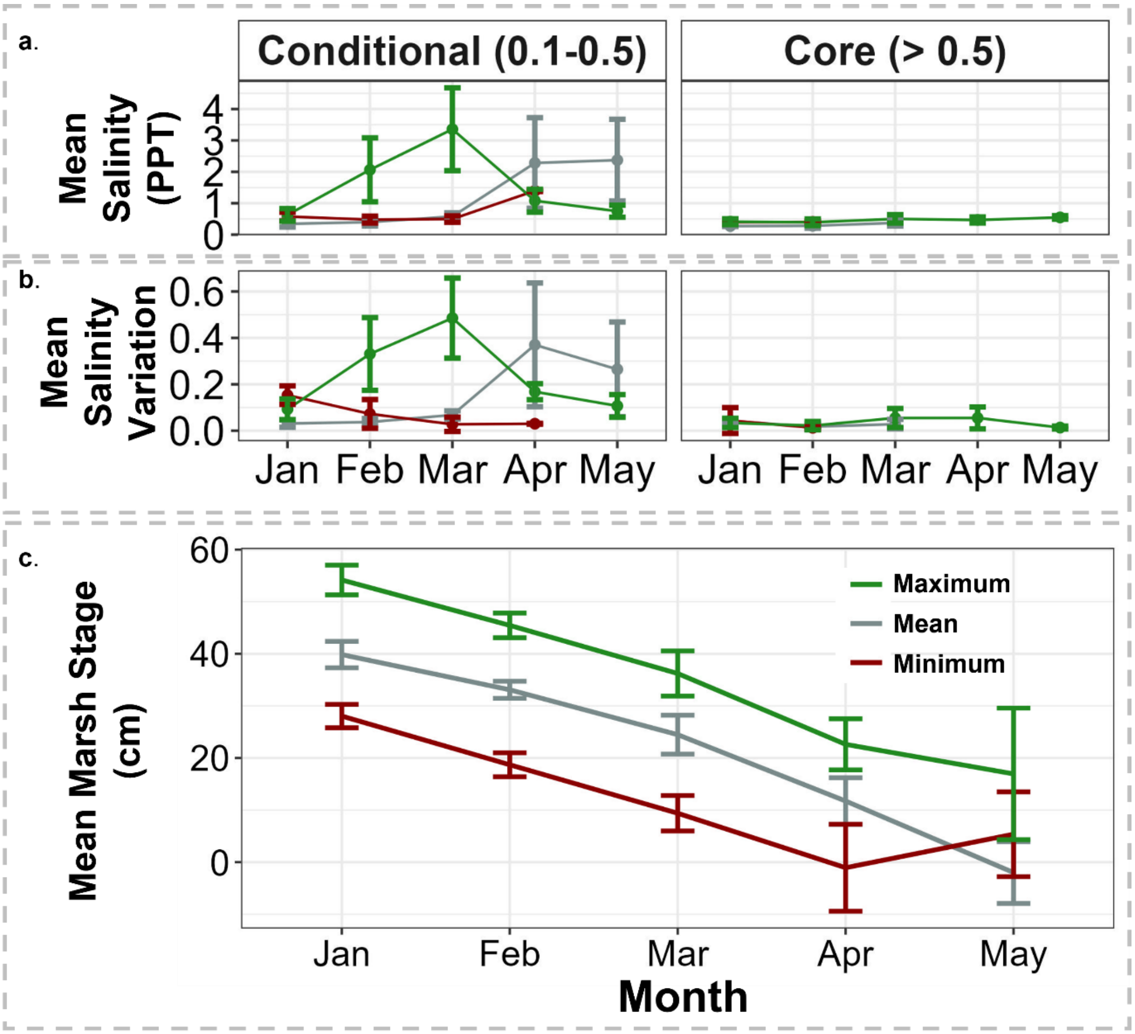


Figure 5. Mean values of top three most important varibles in BRT; mean salinity in PPT (a), mean standard deviation of salinity (b), mean stage of surrounding marsh habitat in cm (c) by year. Mean salinity (a) and standard deviation (b) are separated by habitat classification; left = conditional (grid cells with 0.1 - 0.5 predicted occurrence) and right = core, (grid cells with greater than 0.5 predicted occurrence). Marsh stage (c) is a riverwide varible and held constant under both habitat classifications. Colors represent years; green = Maximum, gray = Mean, and red = Minimum spatial coverage years.



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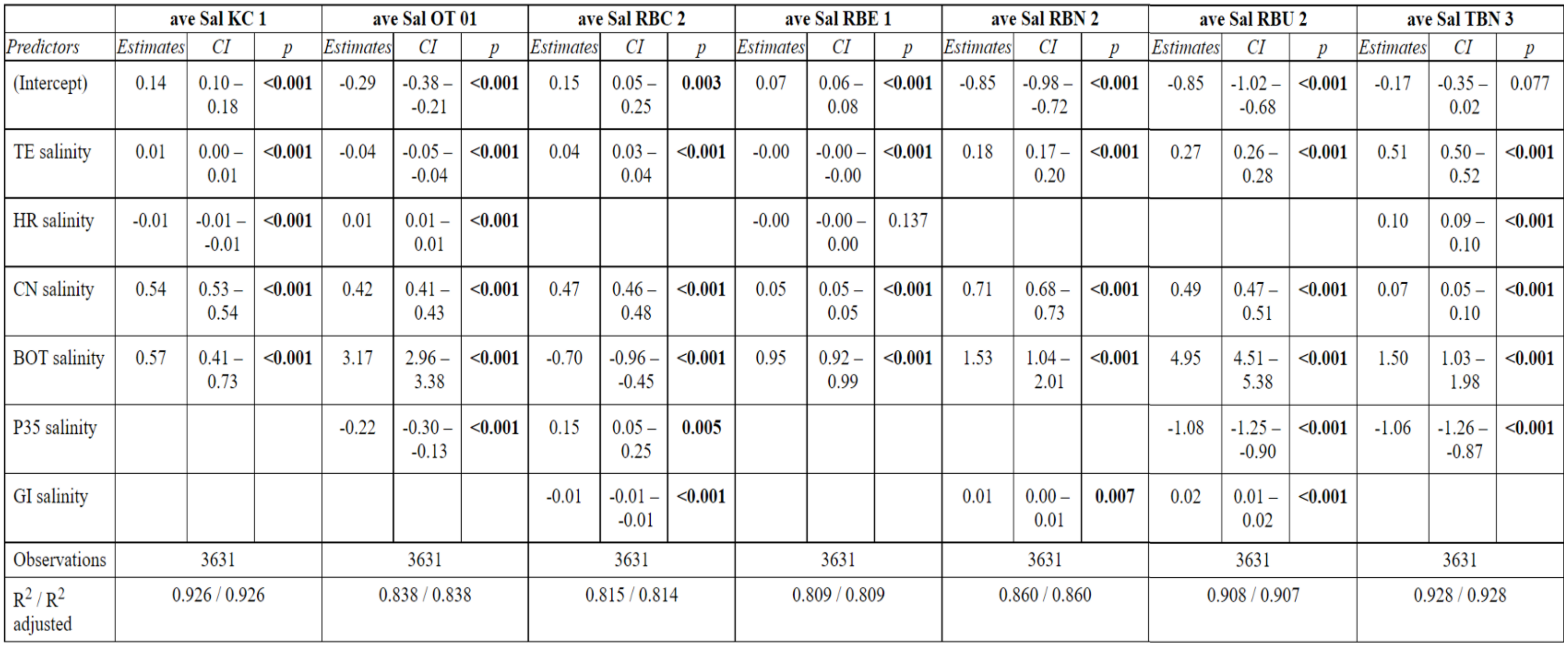
APPENDIX

Table A.1 Summary of acoustic tag metrics by year. A study period summary was included in the las Standard error was used to calculated variability. Since nearly half of the tagged individuals were detected in more than a single year the number of unique tags for the study period does not equal the sum of tags detected each year.

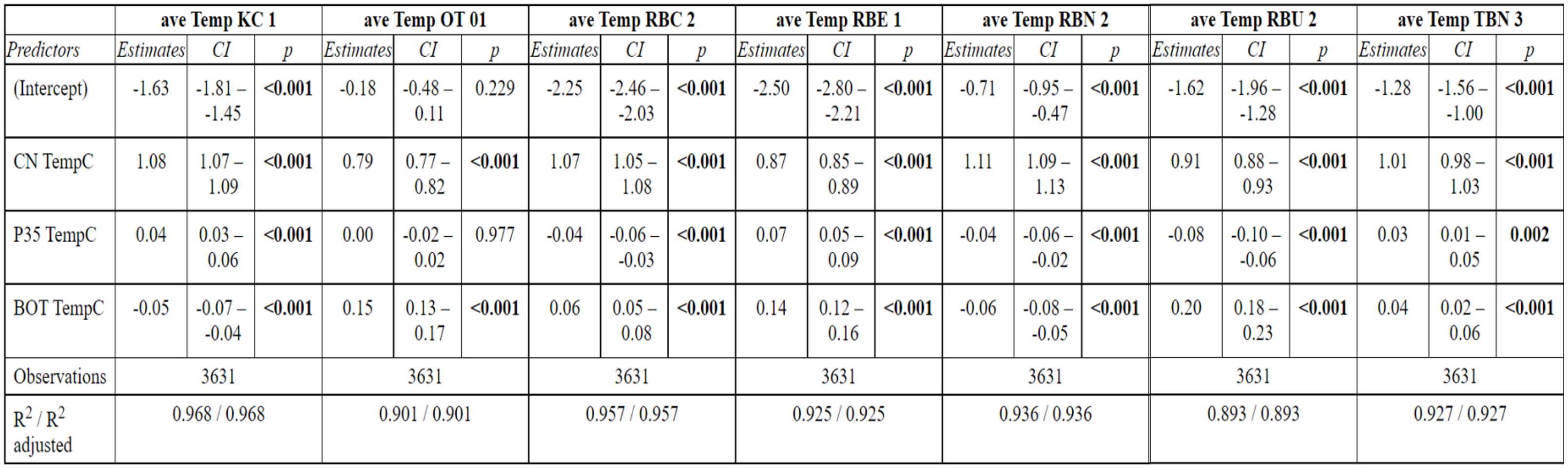
Year	Number of Unique Tags	Mean TL (cm)	Mean Weight (kg)	Mean Days Detected
2014	12	47.5 ± 1.3	1.7 ± 0.2	41.3 ± 6.0
2015	20	46.1 ± 0.8	1.5 ± 0.1	47.2 ± 10.9
2016	13	46.6 ± 1.3	1.5 ± 0.1	101.5 ± 34.3
2017	28	46.5 ± 0.7	1.6 ± 0.1	149.6 ± 29.2
2018	32	47.3 ± 0.8	1.7 ± 0.1	169.6 ± 24.6
2019	30	47.9 ± 0.8	1.8 ± 0.1	165.9 ± 24.5
2020	23	48.9 ± 0.7	1.9 ± 0.1	148.4 ± 32.0
2021	23	49.2 ± 0.5	1.9 ± 0.1	124.6 ± 20.8
Study Period	**101**	**47.5 ± 0.9**	**1.7 ± 0.5**	**118.5 ± 22.8**

Table A.2 Salinity regression model results. Column names; mean (ave) salinity (Sal) HOBO logger station (KC1, OT01, RBC2, RBN2, RBU2, or TBN3). Row names; fixed hydrostation used as predictior.



	ave Sal KC 1			**ave Sal OT 01**			**ave Sal RBC 2**			**ave Sal RBE 1**			**ave Sal RBN 2**			**ave Sal RBU 2**			**ave Sal TBN 3**		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	0.14	0.10 – 0.18	**<0.001**	-0.29	-0.38 – -0.21	**<0.001**	0.15	0.05 – 0.25	**0.003**	0.07	0.06 – 0.08	**<0.001**	-0.85	-0.98 – -0.72	**<0.001**	-0.85	-1.02 – -0.68	**<0.001**	-0.17	-0.35 – 0.02	0.077
TE salinity	0.01	0.00 – 0.01	**<0.001**	-0.04	-0.05 – -0.04	**<0.001**	0.04	0.03 – 0.04	**<0.001**	-0.00	-0.00 – -0.00	**<0.001**	0.18	0.17 – 0.20	**<0.001**	0.27	0.26 – 0.28	**<0.001**	0.51	0.50 – 0.52	**<0.001**
HR salinity	-0.01	-0.01 – -0.01	**<0.001**	0.01	0.01 – 0.01	**<0.001**				-0.00	-0.00 – 0.00	0.137							0.10	0.09 – 0.10	**<0.001**
CN salinity	0.54	0.53 – 0.54	**<0.001**	0.42	0.41 – 0.43	**<0.001**	0.47	0.46 – 0.48	**<0.001**	0.05	0.05 – 0.05	**<0.001**	0.71	0.68 – 0.73	**<0.001**	0.49	0.47 – 0.51	**<0.001**	0.07	0.05 – 0.10	**<0.001**
BOT salinity	0.57	0.41 – 0.73	**<0.001**	3.17	2.96 – 3.38	**<0.001**	-0.70	-0.96 – -0.45	**<0.001**	0.95	0.92 – 0.99	**<0.001**	1.53	1.04 – 2.01	**<0.001**	4.95	4.51 – 5.38	**<0.001**	1.50	1.03 – 1.98	**<0.001**
P35 salinity				-0.22	-0.30 – -0.13	**<0.001**	0.15	0.05 – 0.25	**0.005**							-1.08	-1.25 – -0.90	**<0.001**	-1.06	-1.26 – -0.87	**<0.001**
GI salinity							-0.01	-0.01 – -0.01	**<0.001**				0.01	0.00 – 0.01	**0.007**	0.02	0.01 – 0.02	**<0.001**			
Observations	3631			3631			3631			3631			3631			3631			3631		
R^2 / R^2 adjusted	0.926 / 0.926			0.838 / 0.838			0.815 / 0.814			0.809 / 0.809			0.860 / 0.860			0.908 / 0.907			0.928 / 0.928		

Table A.3 Temperature regression model results. Column names; mean (ave) temperature (Temp) HOBO logger station (KC1, OT01, RBC2, RBN2, RBU2, or TBN3). Row names; fixed hydrostation used as predictior.



	ave Temp KC 1			ave Temp OT 01			ave Temp RBC 2			ave Temp RBE 1			ave Temp RBN 2			ave Temp RBU 2			ave Temp TBN 3		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	-1.63	-1.81 – -1.45	**<0.001**	-0.18	-0.48 – 0.11	0.229	-2.25	-2.46 – -2.03	**<0.001**	-2.50	-2.80 – -2.21	**<0.001**	-0.71	-0.95 – -0.47	**<0.001**	-1.62	-1.96 – -1.28	**<0.001**	-1.28	-1.56 – -1.00	**<0.001**
CN TempC	1.08	1.07 – 1.09	**<0.001**	0.79	0.77 – 0.82	**<0.001**	1.07	1.05 – 1.08	**<0.001**	0.87	0.85 – 0.89	**<0.001**	1.11	1.09 – 1.13	**<0.001**	0.91	0.88 – 0.93	**<0.001**	1.01	0.98 – 1.03	**<0.001**
P35 TempC	0.04	0.03 – 0.06	**<0.001**	0.00	-0.02 – 0.02	0.977	-0.04	-0.06 – -0.03	**<0.001**	0.07	0.05 – 0.09	**<0.001**	-0.04	-0.06 – -0.02	**<0.001**	-0.08	-0.10 – -0.06	**<0.001**	0.03	0.01 – 0.05	**0.002**
BOT TempC	-0.05	-0.07 – -0.04	**<0.001**	0.15	0.13 – 0.17	**<0.001**	0.06	0.05 – 0.08	**<0.001**	0.14	0.12 – 0.16	**<0.001**	-0.06	-0.08 – -0.05	**<0.001**	0.20	0.18 – 0.23	**<0.001**	0.04	0.02 – 0.06	**<0.001**
Observations	3631			3631			3631			3631			3631			3631			3631		
R^2 / R^2 adjusted	0.968 / 0.968			0.901 / 0.901			0.957 / 0.957			0.925 / 0.925			0.936 / 0.936			0.893 / 0.893			0.927 / 0.927		

Table A.4 Common Machine Learning evaluation metrics. Values are given in probabilities. The true positive rate is determined by the model’s ability to correctly classify a positive (presence), true negative rate is determined by the model’s ability to correctly classify a negative (absence), false positive rate is determined by instances where the model erroneously predicted presence, and false negative is where the model erroneously predicted absence. Positive and negative predictive power is the model’s overall ability to correctly predict occurrence (presence/absence).

ML Evaluation Metric	**Score**
Overall diagnostic power	0.89
Area under the receiver operator (ROC) curve	0.98
Correct classification rate	0.94
True positive rate	0.93
True negative rate	0.95
False positive rate	0.05
False negative rate	0.07
Positive predictive power	0.68
Negative predictive power	0.99
Misclassification rate	0.06

Figure A.1 Marginal effect plots for remaining variables in BRT and their relative importance; Julian day 13.8% (a), temperature 12.6% (b), depth 12.5% (c), and standard deviation of temperature 4.5%.

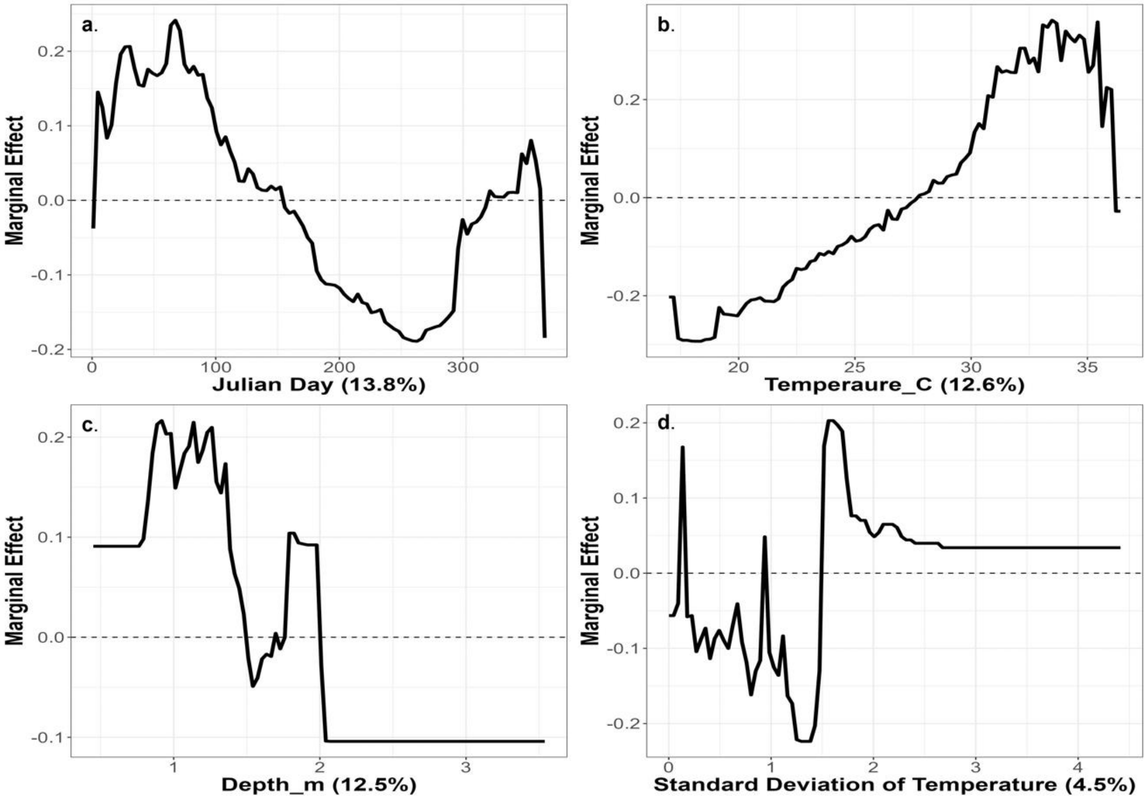


Figure A.2 Total area classified as conditional (10-50%) and core (> 50%) by year. Years selected as Maximum, Minimum, and Mean scenarios are highlighted by colored box, Maximum = Green, Minimum = Red, and Mean = Gray. Values are the cumulative sum for each classification.

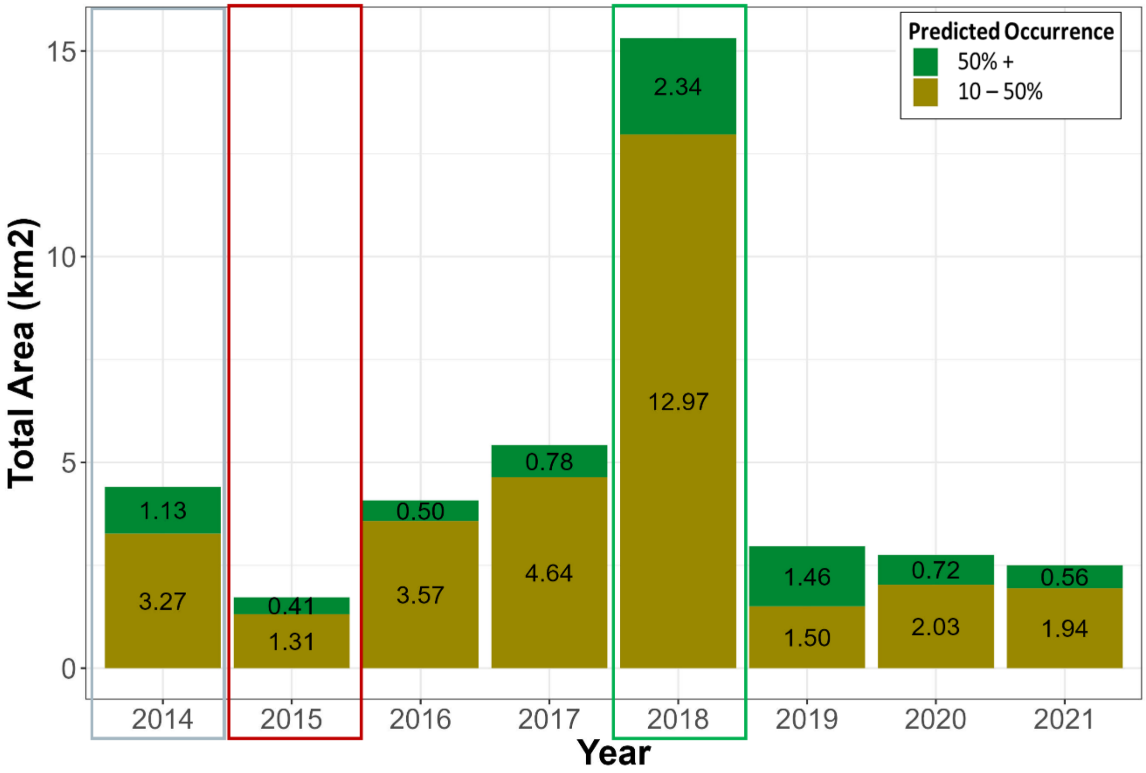
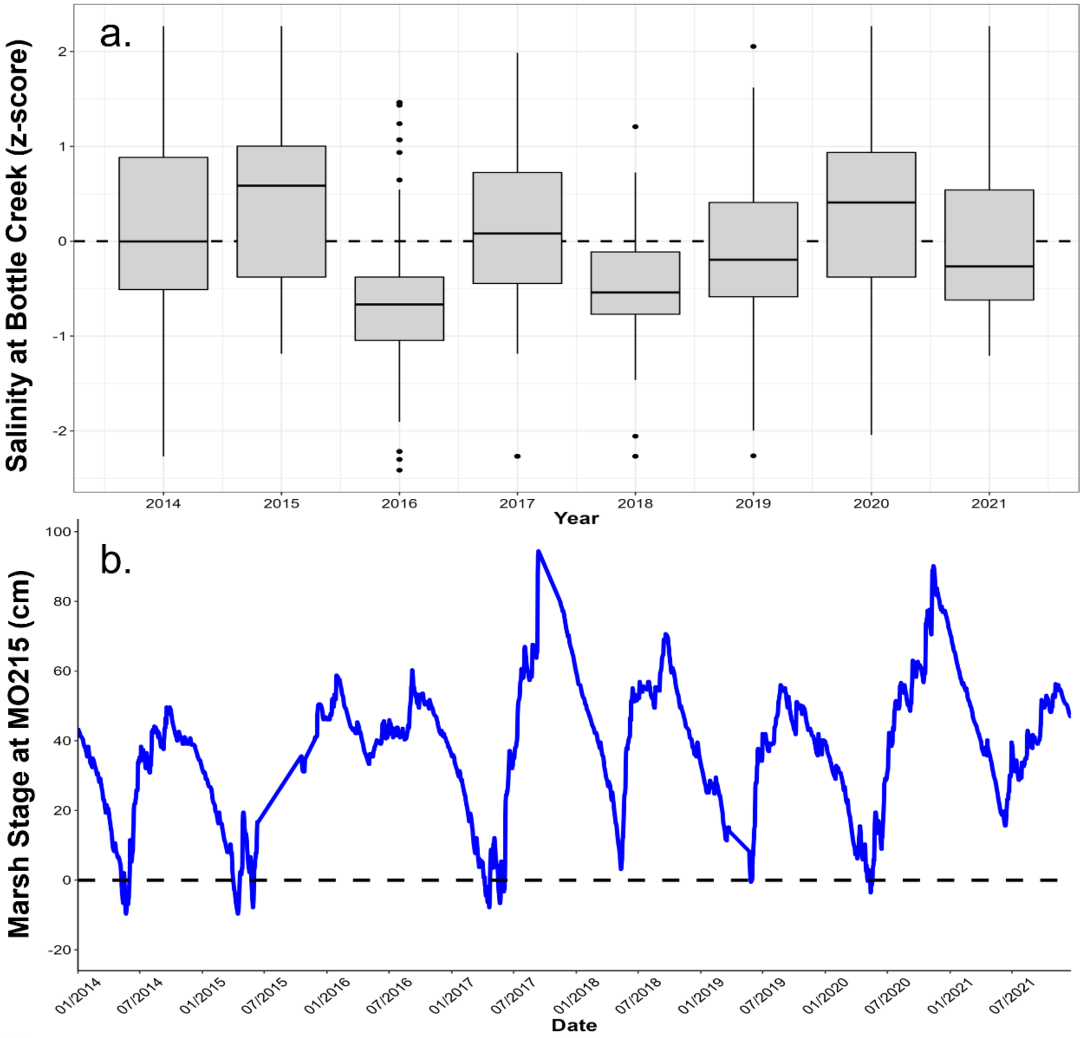


Figure A.3. Study period timeseries of top three most important variables in BRT. For salinity box plots, bars = median, boxes = interquartile range (low = 25th percentile, upper = 75th percentile), whiskers = range in data values. Salinity has been z-scored to show deviation from study period mean (a.). Marsh stage below the dotted reference line indicates a full drying of surrounding marsh habitat (b.). Salinities are elevated during years with severe marsh drying.



CHAPTER III

GENERAL CONCLUSION

Adaptive management of our natural resources is becoming increasingly vital (Henry and Sorte 2022). A deep understanding of how species respond to changes in the environment is critical for developing effective management strategies. Further, a focus on species and environments that are most at risk allows for efficient use of limited resources (Weiskopf et al. 2020). This work used long-term acoustic telemetry and environmental data to better characterize what conditions provide essential dry season refuge habitat for an important recreational freshwater fish species. We recommend water management strategies that aim to ensure the persistence of dry season refuge in the face of climate change.

Our results indicate that the amount of refuge habitat available for Bass in Shark River (SR) is dynamic and is mainly a factor of freshwater availability (marsh stage) and salinity. Our findings suggest managing freshwater deliveries to prevent complete drying of marsh habitat and maintain salinity in the Upper SR below 10 PPT can help preserve vital dry season refuge. Since Bass are considered indicator species, it is likely that managing habitat for them will benefit other freshwater species (Kemp et al. 2019). It should be noted, however, that freshwater fish have higher salinity tolerance than other freshwater species and targets provided by this work may fail to maintain suitable habitat for freshwater invertebrates (Castillo et al. 2018). Future work should focus on evaluating SR as dry season refuge habitat for a variety of species.

Our ability to model future climate change scenarios increases practically daily and future work should take advantage of available resources such as the Everglades Landscape Model (ELM, Fitz et al. 2011). Flower et. al (2017) used ELM to model and evaluate shifts in freshwater habitat for Everglades National Park which helped inspire the work done here (Flower et al. 2017). Future work should focus on expanding climate change scenarios to include ongoing restoration efforts (National Academies of Sciences, Engineering 2021).

Estuaries are experiencing some of the most extreme changes in tidal range world-wide (Talke and Jay 2020). Given that salinity is a main factor determining community composition and distribution in estuaries (Whitfield et al. 2012), changes in tidal range will likely affect many species that rely on estuaries for various life phases. Identifying estuaries that are most at risk due to climate change and anthropogenic disturbances should be the focus of future research. Further, estuaries that act as vital refuge habitat should continue to be evaluated since species that use these refuge habitats are among the most vulnerable and unlikely to have the adaptive capacity to tolerate elevated salinities.

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